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A review of the literature on benefits, costs, and policies for wildlife management

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Abstract; Wildlife management is a source of conflict in many countries because of the asymmetric allocation of benefits and costs among stakeholders. A review of studies on benefits, costs, and policies shows most valuation studies estimate recreational values of hunting, which can range between 13 and 545 USD/hunting day (in 2013 prices). A majority of the studies on costs calculate losses from livestock predation and crop destruction, and show that they can correspond to 40% of profits in the agricultural sector in wildlife rich regions in the US. Most of the studies are carried out for animals in developed economies, in particular in the US. This is in contrast to studies on costs of wildlife, which to a large extent are born by farmers neighboring national parks in developing and emerging economies. However, a common feature of both valuation and cost studies is the exclusion of several costs and benefits items and of indirect effects in the economies, which can be considerable for economies with high reliance on tourism and agriculture sectors. With respect to policy choice, the literature suggests economic incentives for conflict resolutions, where the winners from wildlife compensate the losses, but studies evaluating such policies in practice are lacking.

Key words: costs, benefits, policies, wildlife, review

JEL codes: Q29, Q57

1. Introduction

Wildlife populations, which we refer to as non-domesticated animals, are the sources of both costs and benefits to society. Costs occur from wildlife predation on livestock, destruction of crops, traffic collisions, and transmission of diseases to animals and humans. Values accrue from hunting, recreational activity, food, and other ecosystem services. However, the asymmetric allocation of these costs and benefits among stakeholders is one important source of current threat of wildlife. Approximately 50% of all the mammals worldwide are in decline and 25% are facing extinction because of destruction of habitats for e.g. agricultural purposes (Roemer and Forest, 1996; Woodroffe and Ginsberg, 1998) and illegal hunting (Treves and Karanth, 2003; Pack et al., 2013). For example, in eastern and southern Africa, economic losses due to carnivores' predation on livestock can range from 1 to 25 percent of potential revenue, and carnivores can severely reduce the quality of life (Bulte and Rondeau, 2007). Increases in land use for agricultural purposes can increase this type of costs which is most likely to occur in developing countries (Madhusudan, 2003). However, wildlife also provides a source of benefits and incomes, where hunting and animal watching can generate significant profits (see e.g. Barnes et al., 1999; Hoofer, 2002 for examples).

Another cause for wildlife threat is the lack of efficient policies which would adjust for the asymmetric allocation of costs and benefits and create incentives for wildlife preservation and restoration. For example, when farmers are not fully compensated for the experienced loss due to wildlife while at the same time revenue is generated from the existing wildlife there exists a market failure and an associated allocation problem (e.g. Sommers et al., 2010). However, despite the concern of wildlife management since 1950s (e.g. Gordon 2004) there exists no survey of studies estimating costs and benefits of wildlife, which would give some indication on their relative importance. The purpose of this study is to carry out such a survey in order to investigate the magnitude of the costs and benefits, which animals have been at focus and in which regions, and the method used for the calculations. In addition, we review policies suggested in the literature with the aim of creating incentives for improved wildlife management.

A limitation of the study is that we do not attempt to explain differences in estimates, which would require separate meta-analyses of the cost and benefit components. Instead we aim to identify levels of estimates and eventual differences among studies. Another limitation is the exclusion of studies on efficient wildlife management with no explicit focus on policy design, which usually rests on dynamic optimization of net social benefits with underlying model(s) of wildlife population dynamics. Skonhoft (2007) gives a comprehensive and thorough analysis of different modelling approaches in the economics of wildlife management.

The remainder of this paper is organized as follows; Section 2 gives an overview of studies on the economic values derived from wildlife, Section 3 presents studies on costs associated with wildlife, and Section 4 makes a survey of studies on policy design for wildlife management. The paper ends with concluding remarks.

2. The economic value of wildlife

Since "Waterfowls and Wetlands" by Hammack and Brown (1974) the estimation of the economic value of wildlife received considerable attention in the past (Livengood, 1983). In principle, economists tend to agree that there are two aspects of the total economic value of a wildlife resource; use values and non-use value. The use-values capture values that are associated with the active use of the resource, such as meat value, hunting, fishing, viewing etc. and nonuse-values include all other values outside the use category (Devouges et al., 1983; Fisher and Raucher, 1984; Boyle and Bishop, 1987). Other studies agree that there are use-values but classify non-use values into intrinsic and existence values, which accrue to everyone that has an interest in wildlife (Krutilla, 1967; Stevens et al., 1991). So to say, the "existence"-value relates to the willingness to pay of wildlife-interested individuals for the pure existence or preservation of the resource. However, despite preliminary evidence that the existence-values have an overall bigger share in the total economic value of wildlife, they are often met with skepticism (Brookshire et al., 1983; Stevens et al., 1991; Zawacki et al., 2000).

Estimation of use and non-use values requires different methods; revealed or stated preference methods. The revealed preference methods are used for estimation of use values. They are based on valuation of wildlife through the behavior on markets, directly or indirectly. For example, the value of meat is expressed on the market price of meat, and the hunting value through the payments of hunting fees. However, these methods cannot be used for estimating non-use values which instead rest on stated preference methods where respondents state their value of hypothetical changes in wildlife in some type of questionnaire. These methods usually examine the total value of a resource for an individual including both use and non-use values.

In addition to the estimation of individuals' valuation of wildlife, mainly in terms of value of hunting, a few studies have estimated the effects of wildlife as a sector promoting regional economic development. This is made by constructing social accounting matrices (SAM) and input-output (IO) models which link the wildlife sectors to the rest of the economy and calculate direct and dispersal effects of changes in the demand for wildlife activities.

In this section, we report results from studies estimating individuals' valuation of wildlife, and on effects on the economies. The estimated values are given in U.S dollar (USD) computed at 2013s prices. Included studies are both peer-reviewed and studies in the 'grey literature'. The grey literature on the estimation of value of hunting activity estimates is relatively large (CREST, 2014).

2.1 Use and non-use values of wildlife

The meat value is relatively simple to estimate since, in general, there is a market price on this item which can be used. Estimation of other values, such as recreational values, implies more of a challenge since there is no direct market for this. The revealed preference methods rely on behavior in indirect markets which can be related changes in wildlife condition. The travel cost method (TCM) is one of the most applied revealed preference method. It measures the willingness of hunters to pay for a hunting trip or day, assuming weak complementary, as developed by Mäler (1974). This approach links unpriced public goods to a priced market-good.

That allows inferring statistically on the value of wildlife resources by utilizing hunting related cost, for example travel-, equipment- or license cost. On the other hand, hedonic techniques compare the difference of values between an observable market price and the immediate effects of a non-market good on the market price of the same good. That is, for example, the comparison of the variable "price of land" vs. the variable "price of land in a recreational area" (Whitten and Bennet, 2001). However, outcomes of these studies only refer to "use"-values and are not appropriate to evaluate "existence"-values. Furthermore, the assumption of weak complementary is essential to generate credible results and even though TCM delivers a good and reasonable approximation of the WTP, it is not revealing further conclusions on hunter's preferences, what might be desirable for policy- or non-touristic purposes (Loomis, 2000).

In order to deal with the problems and limitations of TCM, another method is widely spread in the valuation of non-market goods and especially in the valuation of hunting. Contingent valuation methods (CVM) use artificially created hypothetical markets and rely on human decision, so called stated preference models (Loomis, 2000; McFadden, 1986). The idea was first suggested by Ciriacy-Wantrup (1947) and Robert Davies (1963a, 1963b, 1964), who were the first to use interviews with hunters and recreationists to reveal their willingness to pay for a specified recreational area in Maine (Mitchell and Carson, 1989). Since then, CVM has vastly been applied in the valuation of public goods.

CVM include different methods and is therefore a uniting expression for stated preference methods, which also include conjoint analysis and choice experiments. The chosen method depends upon the preferences of the researcher, the topic, the data or another mix of these factors. The most used measures to evaluate the willingness-to-pay or –accept are hypothetical bids that are given from respondents and may take every possible value, which is the so called open-end application (Mackenzie 1990, see application by Sorg and Nelson, 1986; Young et al., 1987). Conjoint analysis evaluates the decomposed attributes of a good with respect to respondents' marginal rate of substitution (Mackenzie, 1990). Choice experiments create bundles of attributes with values that have been predetermined by the researcher and that are chosen from the individuals, to decrease the quantity of choice possibilities and reduce potential biases. Even though conjoint analysis and choice experiments may appear similar in design and organization they experience fundamental differences in consistency and the underlying theory (Louviere et al., 2010).

Studies on the estimation of hunting values have been conducted since early 1970s using all listed methods, but to varying degree (Table 1).

Table 1: Classification of 36 studies estimating value of hunting in different categories with respect to valuation method, animal and region under study, and results in 2013 USD

Valuation method	*Number of studies*	*Periods*	*Animal (number of studies)*	*Regions (number of studies)*	*Hunting value measured in different ways*[4]
Revealed preference methods[1]**;**					
Travel cost method	14	1970-2012	Big game (6), deer (4), elk (3), small game (3)	States in US (13), Australia	13-354/day; 89-255/year; 24-236/trip
Hedonic method	2	1975-1980	Big game (1), deer (1)	States in US	184; 71/deer
Stated preference methods;					
Contingent valuation[2]	15	1970-2006	Small game (1), elk (6), deer (3), moose (3), others (2)	States in the US (12), Canada (1), Sweden (2)	38-175/day; 24-184/trip; 1133/deer; 612/season
Choice experiment[3]	5	1996-2014	Deer (3), Moose (1), Partidge (1)	Finland (1), Spain (1), New Zealand (2), Scotland (1)	125-545/day; 3-455/animal

[1]Table A2 in appendix; [2]Table A3 in appendix; [3]Table A4 in appendix; [4]Ranges or single number are given for estimated values per day, per trip, or totally.

The number of studies, which total to 36, are almost equally divided between revealed and stated preference methods but the TCM and CVM dominate in their respective category. Value per hunting day is the most common measurement for both methods, which can vary between 13 and 545 USD/day. Approximately 80% of the studies are applied to wildlife the US, where they have been conducted by local state agencies. The estimates are mainly used to evaluate a site on its recreational value to either optimize wildlife-management and/or use the findings for touristic purposes.

Most of the studies belong to the non-peer-viewed literature and are mainly reports for local hunting- and fishing-departments. They have not been carried out repeatedly or on the same species in different regions in order to compare results or monitor how environmental factor or hunters perception change over years (except Boman et al., 2011). Some of the studies, such as Young et al. (1987), Donelli and Nelson (1986), Sorg and Nelson (1986) evaluated the hunting experience with both, TCM and CVM approach. Unfortunately, comparability in regard to the results is rarely given. In the studies, some values are rather close (see Young et al. 1987) while others are very different (see Sorg and Nelson 1986, Donelli and Nelson 1986), what is explained by differences in the data and therefore confirm that it could be helpful to determine the factors that influence the hunter's value. Nevertheless, one should keep in mind that most of the studies follow the purpose to derive policy-recommendations on how to use land, for example by creating benefits by opening it for hunters and not to compare the value for hunters in different regions.

Although hunting-values might be difficult to use as time goes by, findings related to preferences might be valid. For example Morton et al. (1994) reports that with increased wildlife abundance and low-hunting pressure, hunters are willing to pay up to 4.5 times for a hunting trip. Fried et al. (1995) state that, under the assumption of certainly bagging an elk in Oregon, the WTP is 457.54 $ (2013-USD) per elk, which is 5,5 times more than the calculated hunting value per day. Furthermore, Livengood (1983) and Loomis et al. (1988) report almost double WTP/recreational value per trip if individuals may expect shooting a high-ranked trophy elk bull or white-tailed buck. The fact that hunters spend exceedingly more if they bag a desired trophy gives rise to further literature dealing with the utilization of trophy hunting as touristic activity. Lewis and Alpert (1997) as well as Lindsey et al. (2006) evaluate the effect of trophy hunting in Africa and attest great possibilities of reducing bushmeat hunting, improving wildlife conservation and gaining rural income of structural weak regions, if local communities are involved in policy-measures. If hunters know, that fees from trophy hunting benefit local communities, they are willing to spent even more for a trip, than just for bagging a desired trophy. However, this is rather a question of policy measures than one of valuing game hunting.

Table 1 also reveals an interesting shift in the choice of valuation methods, which is also applicable to the valuation of other environmental changes. Choice experiments (CE) have increasingly been used in environmental economics and for cost-benefit analysis, especially in the valuation of non-market-goods (Adamowicz et al., 1994; Adamowicz et al. 1998a; Boxall et al., 1996; Layton and Brown 2000). CE's are also a form of contingent valuation/stated preference models, in which respondents have the opportunity to choose one alternative from a hypothetical set of choices. This alternative includes different attributes and characteristics as well as a monetary value (closed-end-application). By choosing one alternative the respondent makes a trade-off between different values of some attributes of a choice set and by doing so, repeatingly gives values to the characterizing attributes of a good. Technically, the main advantages are the reduction of biases that often concern CVM and the possibility of testing the model for internal consistency. Nevertheless, the greater amount of information that can be extracted from the answering individual is probably the main reason why this method lately experienced an increase in popularity (Alpizar et al., 2003).

All of CE studies but Horne and Petäjistö (2003) evaluate hunting trips. Bullock et al. (1998) and Delibes-Mateos et al. (2014) conclude that quality of game is the most important factor for hunters willingess-to-pay. Evaluating red deer hunting in Scotland and red-legged partridge hunting in Spain, the experience/sporty aspect of hunting is paramount for most of the hunters. Therefore, hunters prefer shooting wild red-legged partridges instead of farm-reared ones, that mostly show significant deficiencies in their abilities to fly, spot predators etc. and thus are an easy target. For every wild partridge hunters would pay more than 20 times than for one farm-reared (Mateos-Delibes et al., 2014). Furthermore, some hunters prefer to hunt hinds instead of red-deer stags, due to their higher alertness and the resulting higher demands on the hunter. Trophys are generally not that important; however, the certainty of shooting a high trophy stag is increasing WTP's quite significantly (Bullock et al., 1998). The same is reported in Kerr and Abell (2014a/b) on Sika and Himalayan Thar, where trophies play a significant role for experienced hunters. They classify hunters by defining criteria in generalist, experienced and local hunters and describe respective WTP's for attributes like access to the hunting site, environment etc. According to the studies, the WTP for hunting trips is reduced the closer a hunter lives to a site.

Only Horne and Petäjistö (2003) evaluate the value of hunting moose in Finland from the perspective of the landowners with substantial forestry on their land. They show that despite good benefits from moose hunting a reduction of the population would be appreciated, even if landowners would be even more financially involved on the sales of hunting licenses.

2.2 Regional economic effects

In recent times the use of revealed and stated preference methods has decreased relative to that on the estimation of total economic impacts of wildlife. Associated methods for estimating regional economic effects of hunting have developed. The most simple way of measuring this is to assess expenditures for hunting, such as transport and equipment. These expenditures will give rise to second order effects in a region, where sectors with immediate benefits from hunting activities impact other sectors through their demand for deliveries. In order to capture all effects of hunting activities social accounting matrices (SAM) are constructed and input-output models (I-O) are employed for the evaluation of total economic impact in terms of economic multipliers which show the dispersal effect of hunting activities in a region. Studies using this method estimate direct and indirect effects related to the first and second round of spending, but also for leakages in the region studied (Burger et al., 1999; Pickton and Sikorowski, 2004; Grado et al. 2007; The U.S Department of Interior, 2011; CREST, 2014). The second round of spending can be important for the economic impact. For example, Grado et al., (2007) estimated that 80.78 million USD was spent in the state of Mississippi by non-resident hunters i.e. hunting tourism. Spending per hunter-day is usually higher for non-resident hunters than resident hunters. Pickton and Sikorowski (2004) estimated that average spending by resident hunters ranged approximately between 32-35 USD while non-resident hunters spent on average 157-300 USD.

Table 2: Classification of 9 studies estimating regional economic benefits of wildlife in different categories[1]

Method	*Number of studies*	*Period*	*Animal (number of studies)*	*Regions (number of studies)*	*Estimated benefit in 2013 USD*
Expenditures	7	1991-2012	Leopard (1), bear (1), wildlife (5)	US (2), Africa (4), Europe (1)	0.03-34.8 billion
Social accounting matric	1	2001-2005	Deer	US	0.8-1.3 billion
Input-output analysis	1	1991	Northern Bobwhite	US	0.3 billion

[1]The studies are listed in Table A5

Hunting tourism, or trophy hunting, is a common activity, especially in developing countries and thus provides an inflow of capital to a region. Hofer (2002) estimated that 20-30% of the European hunters go abroad occasionally to hunt and together they generate 70-87 million USD in revenue which stays in the visited country and is not subject to leakages. Table A1 shows the annual direct expenditure in USD for a selected number of countries extracted from Hofer (2002). The figures show that the share of GDP can range between 0.01% (Spain) and 0.47% (Great Britain). Domestic expenditure on hunting activities thus generates significant economic value according to these figures.

Trophy hunting is a relatively large activity in sub-Saharan Africa and 23 countries in the region permit trophy hunting (Lindsey et al., 2006). The southern parts of Africa (Botswana, Mozambique, Namibia, South Africa, Swaziland, Zambia, and Zimbabwe) have 88 % of the trophy hunters but South Africa has the largest trophy hunting sector (Lindsey et al., 2006). The profitability of trophy hunting depends on the abundance of game, and according to Lindsay et al., (2006) there is a negative trend in the trophy hunting business due to animal depletion. This is most likely due the common reasons threatening wildlife overall, e.g. population growth and increased land usage, and over-exploitation. Hence there might be a window where conservation and hunting activities can both profit from sustainable use of wildlife as a natural resource.

3 Cost of Wildlife

Wildlife, specifically carnivores and herbivores can inflict economic damage to society, in particular to farmers and forestry. The most common type of damage occur either through losses of crops or predation on livestock (Baker et al., 2008) where carnivores are usually responsible for predation while herbivore cause the majority of the damage to crops. A majority of the literature estimating costs of wildlife up to 1995 focused on one species, one crop or one region, (Conover et al., 1995), indicating that the complete picture of the cost of wildlife is not known. Another common source of cost is wildlife-vehicle collisions, with the inclusion of the value of human life in the estimate total values can be substantial. Other types of damages exists, such as, spreading of diseases, but the literature estimating these costs is small and will only be mentioned briefly.

Similar to the estimation of values of wildlife, costs of wildlife can be classified into direct and indirect, where the direct costs refers to damages suffered by, e.g. farmers from the killing of livestock. These direct effects may spread into the economy by the reduced supply of livestock, which can affect the prices of livestock and thereby other sectors than agriculture. The literature applies three different methods for calculation of direct cost; questionnaires to stakeholders, analysis of compensation payments, or, for livestock predation, post mortem examination of carcasses i.e. the market value of the lost animal (Baker et al., 2008). All methods comes with different types of advantages and disadvantages, e.g. all carcasses are not always found, implying that farmers cannot be fully compensated, or when using survey methods farmers can exaggerate the economic impact. Market values of crops or livestock do not account for the costs of the production for these goods, or for preventive measures taken for protection against predation. The valuation methods presented in Section 2 can also be used for assessing costs of wildlife by means of questionnaires to stakeholders. The same methods are also used for assessing indirect effects, input-output analysis and partial or general equilibrium methods.

3.1 Predation on livestock

Predation by carnivores on livestock is common and a source of many conflicts, and can cause great damage to individuals. With the increased land use of humans the human-carnivore conflict is increasing and predation on livestock is the most common issue (Abaya et al., 2011). It is important to keep in mind that not all lost animals are due to predators, approximately 1.5% of all animals held are lost to predators (Baker et al., 2008). For ranging animals losses can depend on other factors. Losses experienced due to predation tend to vary significantly over time and space i.e. even on the same farm the losses due to predation can vary substantially (Treves et al., 2004; Baker et al., 2008). This could imply that differences or changes in husbandry and management policies of carnivore may affect the behavior of predators (Baker et al., 2008), however there is no abundance of before and after studies when policies or husbandry has changed on purpose. A probable explanation is that increased predator or carnivore density can increase the amount of predation (Yom-Tov et al., 1995), however the relationship does not always have to be straightforward (Baker et al., 2008). One documented change in husbandry practice that has proved to increase the predation rate is to increase the distance between livestock and human activity (Ciucci and Boitani, 1998).

The body of literature on the estimation of costs of livestock depredation is relatively substantial, but as Conover et al., (1995) acknowledged is mainly focused on one species, livestock or one region. Regions studied varies, there exists studies both from developing and developed countries, although the first can be found in the U.S. Common to most applications is the use of market value of lost livestock as an indication of the direct cost. Estimates of other costs for the farmer and indirect effects in the economy are less frequent (Table 3).

Table 3: Classification of 20 studies estimating costs of lost livestock from wildlife predation in different categories[1]

Method	*Number of studies*	*Period*	*Animal (number of studies)*	*Regions (number of studies)*	*Estimated cost in 2013 USD*
Direct cost[2];					
Market value of livestock	12	1975-2010	Carnivores, hyena (3), felines (2), leopard (2), jackal, wolf (2), bear (2), cats, dogs	US (2), Asia (5) , Africa (2), Europe (2), South America (1)	18-515/household /year 0.01-1.5 million
Compensation payments	2	1992, 2005	Dogs (1), wolf(1)	Africa (1), Europe (1)	0.2, 0.5 million/year
Other[3] farm cost than lost livestock	4	1992-2013	Carnivore (3), wolf (1)	Europe (1), US (3)	0.23-4.3 mill 6620/household/year
Direct and indirect costs[4]	2	1975, 2004	Coyotes (2) and others (1)	US (2)	3.1, 16 mill

[1]The studies are listed in Table A6; [2]Costs at the farm level; [3]Other costs are e.g. costs of prevention measures and impacts on the livestock health, [4]Indirect costs are dispersal effects in the economy; [4]Ranges or single number are given for estimated costs per household, year, or totally.

The direct economic impact on sheep due to livestock depredation has received a lot of attention; Taylor et al., (1979); Mishra, (1997); Ciucci and Boitani, (1998); Asheim and Mysterud, (2004). The responsible predator varies among regions, but a common feature is that one predator is responsible for a large share of all incidents. For example, coyotes were responsible for 94% of the attacks in southwestern Utah during 1972-1975 (Taylor et al., 1979), and 99.6% of the incidents in central Italy were conducted by dogs or wolfs (Ciucci and Boitani, 1998). Studies calculating the economic value by using market value of the lost animal e.g.; Yom-Tom et al., (1995); Mishra, (1997); Butler, (2000); Rao et al. (2002); Madhusudan, (2003); Ikeda, (2004), will not manage to capture the true cost of the loss. In order to fully grasp the economic loss to farmers the relative wealth should also be considered, preventive measures taken, and production capabilities (Baker et al., 2008). Asheim and Mysterud (2004) estimated the total economic cost due to depredation on sheep in Norway, and managed to divide the cost into what maturity *stage* the sheep was in and what the extra cost constituted of. The separation of costs was as follows; consequential cost lambs (10.1%), consequential costs ewes (1.1%), value of lost animal (77.3%), extra labor cost (11.5%). From Asheim and Mysterud's study it is evident that the experienced loss mainly constituted from the economic value of the lost animal.

Identification and estimation of other costs associated with predation on livestock have been acknowledged in recent years and the extent of the economic impact is being investigated (Howery and DeLiberto, 2004; Laporte et al., 2010; Steele et al., 2013; Ramler et al., 2014). The focus of these studies have been on the reintroduction of wolf in different areas in the U.S. Costs associated as indirect costs of depredation are decreased weaning weights, decreased conception rates, reduced weight gain and increased livestock sickness (Steele et al., 2013). For example, Laporte et al., (2010) identified that wolf predation does affect the behavior of both cattle and elk. Furthermore, the threat of predation can reduce the forage efficiency and thus affects the livestock weight gain and physical condition (Howery and DeLiberto, 2004). It follows that predation related stress and injuries increase the risk of livestock becoming ill which can increase veterinary costs (Ashcroft et al., 2010). Farmers subject to depredation can experience extra cost for labor, repairing equipment, searching for lost animals or checking animals (Asheim and Mysterud 2004; Lehmkuler et al., 2007; Sommers et al., 2010;).

Agricultural producers have for a long time been aware of these other costs but relatively little is known regarding the severity (Laporte et al., 2010). Rashford (2010) estimated that the indirect economic impact could be equal or greater than the direct cost of predation, and found that a 5% decrease in weaning weights could potentially reduce the profits of a ranch with 40%. Steele et al., (2013) showed, using data from the Rocky Mountain region in the U.S. that the effect of decreased weaning weights reduced the gross margin by 27% for cattle farmers included in the sample due to wolf predation on cattle, i.e. a lesser effect than the one reported in Rashford (2010). Disease and sickness had only a minor negative effect on the gross margin, 1-2% (Steele et al., 2013). This further implies that financial significance of depredation issues. Farmers in the region receive a 7:1 compensation ratio for lost cattle to account for unverified losses, thus, in order to be fully compensated and include the indirect cost, the ratio would have to be between 18:1 and 24:1. These unidentified and rarely acknowledged costs by compensation programs could reduce tolerance levels for carnivores (Laporte et al., 2010). Ramler et al., (2014) estimated the effect of wolf predation on the average calf weight for 18 Montana ranches. Their result proved that for farms which have a verified incident of wolf predation, the average calf weight decreased with 3.5

Table 3 also shows the relatively few studies estimating indirect effects. In order to fully capture the effect of predation on livestock Jones (2004) argued that the I-O models are appropriate and allow for economic multipliers and analysis of the second round of spending, instead of only capturing the direct loss associated with the first round of spending. Jones (2004) estimated the direct loss due to predation on sheep in 10 different production regions in the U.S to be 21.3 million USD while the indirect loss, as in the second round of spending, amounted to 16 million USD. If the indirect loss can constitute approximately 76 % of the direct cost, it is necessary to analyze the second round of spending.

Although quite many studies apply market values for assessing direct costs, there has been a shift over time in regional application. In recent times focus of research has shifted towards the effect of depredation in developing countries and usually in and around national parks or animal refugees (Mishra, 1997; Butler, 2000; Rao et al., 2002; Madhusudan, 2003; Ikeda, 2004; Michalski, 2006; Gusset et al, 2009; Tamang and Bara, 2008; Abaya et al., 2011), which can be seen more precisely from Table A4. In these cases it is clearer that the farmers' cost is more directly connected with management strategies for wildlife. Common to these studies is the reporting of the loss as annual economic loss per household, and sometimes as a share of annual income. The findings suggest that the economic loss can at an aggregated level be neglected however the losses to an individual household can represent a great share of the annual income, and threaten the households' opportunities to provide for themselves (Abaya et al., 2011). Ikeda (2004) for example reported the loss to yak farmers in the Kanchenjunga conservation area in Nepal. The annual loss ranged from 68.71 USD to 515.34 USD. It followed that the average loss amounted to 29% of the households' pastoralism-related income (Ikeda, 2004). Madhushudan (2003) estimated that the households of Bhadra tiger reserve in south India lost approximately 12% of their livestock only to large felines, while only receiving compensation for 5% of the losses. Mishra (1997) reported losses to approximately one-half of the annual GDP per capita for households in the Kibber Wildlife sanctuary in India due to depredation on goats and sheep. The severity of the losses experienced also depends on the herd size, implying that if a household has a greater herd size, farmers can continue with their business even though they lose livestock to depredation (Ikeda, 2004; Michalski, 2006). Hence, lesser-of households are relatively more affected by depredation.

3.2 Damage to crops and forestry

Damage to crops or forestry is commonly caused by herbivores and in some cases birds or bears, but compared with costs of livestock predation, there are relatively few studies on costs of the damages on crops and forests. Similar to calculation of costs of livestock predation, the two most commonly applied approaches for calculating costs of crop and forest damage have been to send questionnaires to famers to elicit their perceived loss and to use market value of the of actual crop or forestry lost (Table 4).

Table 4: Classification of 11 studies estimating direct costs of damage to crop/forest from wildlife in different categories[1]

Calculation method	*Number of studies*	*Period*	*Animal (number of studies)*	*Regions (number of studies)*	*Estimated cost in 2013 USD[2]*
Market value of lost crops	3	1963-1999	Wildlife (1), wildboar (1), elephants(1)	US (1), Asia (2)	144/household/year; 0.01-4800 million
Compensation payments	2	2000-2010	European bison (1), wildboar (1)	Europe (2)	0.3, 38 million
Perceived loss by farmers	6	1985-1996	Deer and coyotes (3), wildboar (1), Canada geese (1), black bear (1)	US (6)	0-2630/household/year; 2.3-142 million

[1]The studies are listed in Table A7; [24]Ranges or single number are given for estimated values per household/year, per year, or totally.

Approximately half of the studies listed in Table 4 use enquires to stakeholders and the other half market values or compensation payments for estimating crop losses, and wildboar and deer have been the targeted animal for a majority of the studies. We can also note from Table 4 that 7 of the 11 studies are applied to US. It is difficult to compare the results, a major reason being the lack of common denominator. Four studies express costs per household and year, which show considerable differences. Madhusudan (2003) calculate costs of 144 USD/household/year from elephants in India. The other estimates are made for animals in the US; costs of deer and coyotes Conover et al. (1994) range between 0 and 1444/household, cost of black bear to 2038 (Garshelis et al. 1999, and costs of deer ranges from 1112-2630 (Vogel 1989). It is interesting to note that even though 55% of the respondents in Conover (1994) answered that the losses they perceived were too high, most of the farmers reported that they place a high value of having wildlife on their property. Some even actively provided shelter for wildlife.

Similar to livestock predation, damage to crops occur for farmers close to or inside national parks or animal refuges. Farmers in these regions sometimes blame the park management's conservation strategies for their losses (Wang et al., 2006). One example is the restoration of white-tailed deer population in the U.S. which increased the population above the biological carrying capacity level, which created losses for the farmers in the range of 60-142 million USD annually (Vogel, 1989). Hofman-Kaminska and Kowalczyk (2012) examined cost of depredation on crops by the European bison, which is a protected species, in northeastern Poland. During a ten-year period (2000-2010) the total compensation cost was approximately 274 000 USD.

3.3 Traffic damage and other wildlife costs

The cost of vehicle-wildlife collision consists of injuries and fatalities, repair and damage costs. Other costs associated with wildlife are wildlife related diseases, and economic loss to metropolitan households (Conover et al., 1995). Wildlife is a host to several diseases which can be transmitted to humans or livestock (Baker et al., 2008). Compared with estimates of costs of livestock predation and crop losses, there are relatively few studies on costs of collision and transmission of diseases, and they are mainly carried out in developed countries (Table 5).

Table 5: Studies on costs of traffic accidents with and disease of wildlife, cost in 2013 USD

Study	Study period	Animal	Region	Cost type	Cost
Witmer and DeCalesta (1991)	1990	Deer-vehicle collision	Pennsylvania	Repair costs	61.73 million
Conover et al. (1995)	1991	Deer-vehicle collision, Deseases	U.S	Repair costs, value of livestock	1620 billion, 4410 billion
Wywialowski (1994)		Wildlife desease	U.S.	Effect on agriculture	665 million
Bissonette et al. (2008)	1996-2001	Deer-vehicle collision	Utah	Total economic impact	48.5 million (annual: 8.1 million)
Knobel et al., (2005)	2004	Rabies	Africa, Asia	Human health	695 million
Häggström-Svensson et al. 2014	2012	Wildboar collision	Sweden	Repair cost	10 million

All studies on traffic collisions calculate costs in terms of repair costs, and the estimates show large differences. Bissonette et al., (2008) also included fatality and other costs when conducted a study of deer-vehicle collisions in the state of Utah, U.S. The total annual cost was estimated to 8 million which was divided into different components; human fatality (55%), vehicle damage cost (39%), loss of deer (6%) and human injury cost (2%). It is evident that the total cost is heavily dependent on the valuation of human life.

Three of the studies listed in Table 5 have estimated costs of wildlife related diseases. Two of them are applied to US but show quite different results. Conover et al., (1995) give a total estimate of approximately 4.4 billion USD and Wywialowski (1994) estimated the total wildlife related diseases as perceived by farmers in the U.S. to be 665 million USD. Knobel et al., (2005) estimated the annual economic cost for rabies in African and Asia to be 695 million USD and resulted in approximately 55 000 lives lost. The cost of rabies is almost non-existent in the developed world e.g. 8.7 cases per year have been reported in Europe during the period 1995-

2004 (Barker et al., 2008). Knobel et al., (2005) estimated that the amount of livestock lost is approximately 11 000 and 21 000 animals yearly respectively, compared to less than 2000 animals lost in the Europe per year.

4. Wildlife policies

Wildlife policies are typically motivated by either the wildlife species imposing an externality of some kind, e.g. on agricultural or forestry production, or by the species providing ecosystem services that have public good characteristics, e.g. if the species is threatened and its preservation is considered highly valuable. Wildlife policies include, e.g., regulation of hunting rights and management, wildlife damage compensation, economic incentives for hunting and wildlife damage prevention, and voluntary policy instruments for wildlife preservation. À priori, one would expect the choice of policy to differ among wildlife species depending on the private and public benefits and costs that it provides, and on the species behavior, e.g. whether it is sedentary or migratory. In this section, we will first review the economic literature with respect to command-and-control policies related to regulation of ownership or species management. This is followed by a discussion of the literature on wildlife damage compensation. Finally, we investigate the literature on the use of policy instruments for abatement of wildlife damages.

4.1 Command-and-control policies

The literature on regulation of wildlife typically deals with one of three alternative types of policies; *i)* ownership rights to hunting, *ii)* regulation of the hunting of species, and *iii)* spatial policies for wildlife[1]. With respect to the first type, it is argued that the provision of private

[1] We choose to include only studies which extend policy issues beyond the privately optimal management of a single species with zero externalities.

property rights to wildlife on private land provide economic incentives for conservation of rare species, as it permits property owners to market wildlife in various ways. Yet, such rights are not always sufficient to achieve habitat and hence species preservation (Tisdell, 2004; Rasker, Martin and Johnson, 1992). One reason is that the preservation of species is not always privately profitable, as is the case, e.g., if species are mobile, or their values mainly consist of non-use values. Tisdell (2004) takes as an example the decision to give land owners in Australia the right to harvest eggs from a protected bird species, the red-tailed black cockatoo, but where this did not lead to improved habitat management as was first hoped for. Also pessimistic about the potential of strengthened ownership rights, Gibson and Marks (1995) conclude that strategies to redistribute wildlife benefits to local communities in Africa have been little effective in preventing poaching, and the impact on poaching that can actually be observed can merely be attributed to the increased enforcement effort.

Rasker, Martin and Johnson (1992) argue quite differently that creation of private property rights to wildlife has been successful to protect endangered species in Africa. They still find that a combination of fee hunting, game ranching, and improved public ownership could work better in North America. Using game theory, Johannesen and Skonhoft (2005) analyze integrated conservation and development projects (ICDPs), which are frequently set up in African countries. They assume there are two different stake-holders, a park manager that benefits from tourism and hunting, and a group of local people which benefits from illegal hunting but also bears the cost of wildlife damage. Results show that a reallocation of property rights, where the local group is entitled to sharing benefit from park management through a monetary transfer, can increase wildlife conservation but need not imply increased welfare to the local group, and need not imply that the socially optimal outcome is achieved.

Several studies analyze the role of ownership rights with a focus on species that are not threatened, but hunted for their meat and recreational values. Skonhoft (2005) compares socially optimally management of moose, which is a migratory species, to the market solution where each landowner pursues his or her self-interest, and has the right to decide on hunting of species present on their land. He argues that the optimal management approach is more similar to Scandinavian management while the market approach is more similar to management in North America. In Skonhoft and Olaussen (2005), the analysis is extended to account for alternative schemes for seasonal regulation of hunting, and in Nilsen et al. (2009) it is extended to account for differences in mortality rates between subpopulations, due to the wolf presence. Also, using a stage-structured model for moose, Olaussen and Skonhoft (2011) show that a system where private landowners have the right to decide on moose harvesting leads to suboptimal outcomes, due to the externalities in terms of moose-vehicle collisions.

One way to create ownership rights to hunting is through the introduction of markets for hunting licenses or leases. Whereas several have investigated demand and supply for hunting (Sun et al., 2005; Poudyal et al., 2012) fewer have considered how well these markets work, e.g. depending on the existence of market power. Chen and Skonhoft (2013) investigate the impact of land owners' market power on the market for moose hunting licenses, acknowledging that moose provide benefits to hunters and cause forestry damage to land owners. They show that migration of moose is not the only source of interdependence between decisions taken by different landowners, but market power with regard to hunting licenses in combination with moose damages to forest crops can have a considerable impact on land owners' management choices.

The second type of policy, regulated hunting, is commonly applied where threatened species are protected by law, and thereby given a chance to re-establish. This implies that ownership rights to hunting are withdrawn from private agents, and are reserved for the government. Skonhoft (2006) investigates the positive and negative economic consequences of such protection, and subsequent re-colonization, of wolf in Scandinavia, taking into account its direct impact on moose hunting, and secondary impacts on, e.g., moose-vehicle collisions and forest browsing. The results showed that the effect on social net benefits are much affected by the cost of vehicle collision, where increased predation provides benefits by reducing the moose population and, hence, decrease the number of collisions.

With respect to third type of command and control policy, the creation of wildlife conservancies is a commonly used way to protect wildlife within a limited area. Taking into account tourism income, investments, meat and live game value, and employment opportunities, cost-benefit studies have been made for specific conservancies in, e.g., Namibia (Barnes, MacGregor, and Weaver, 2002) and Kenya (Norton-Griffiths, 1996). Also building on cost-benefit analysis, Barnes and Jager (1996) argue that due to scale economies, there are economic incentives for private land-owners to group together and form large wildlife conservancies, but that transaction costs can pose on obstacle. Different to these studies that focus on the costs and benefits which occur within a given conservancy area, Boman et al. (2003) calculate the optimal spatial distribution of wolf in Sweden, taking into account benefits and costs of wolf presence as well as wolf migration. They show that the optimal distribution implies higher wolf densities where there is little competition between wolf and human hunters for common game, such as moose, and small costs of depredation on rein deer, and suggest that their results can motivate a spatially differentiated policy.

4.2 Wildlife damage compensation programs

Compensation programs are applied all over the world and imply that pastoralists and farmers affected by wildlife damages are given the right to compensation for killed and injured animals, and damages to crops (Bulte and Rondeau, 2005). In particular, compensation programs are relatively cheap to implement in areas of poverty, and therefore popular among conservationists and governments (Bulte and Rondeau, 2007). The decision to institute a compensation scheme can be motivated by wildlife damages threatening the livelihood of agricultural producers while at the same time, abatement methods are considered too damaging to wildlife, or by damages being caused by a highly valued species (Yoder, 2000).

Compensation schemes differ with regard to the level of compensation in relation to costs incurred. In some cases, compensation programs payouts are greater than the market value of the lost animal in order to make up for animal losses that cannot be verified (Nyhus, 2003). Often, however, compensation only covers the verified losses, implying that only part of the depredation cost is covered (Sommers et al., 2010). No existing compensation program account for costs for reduced productivity due to, e.g., increased stress among the livestock, and rarely, costs for increased labor time for the pastoralist or farmer are covered. A couple of studies indicate that these other costs are comparable to, or even larger than, the direct costs of wolf depredation for beef producers (Steele et al., 2013; Ramler et al., 2014), whereas Asheim and Mysterud (2004) estimate that productivity and labor related costs in Norwegian sheep farming are approximately one fourth of the direct cost for killed and injured animals.

Many studies point out that compensation programs do not only constitute a transfer of ownership rights and wealth, but also create economic incentives which affect the behavior of agents (Yoder, 2000; Bulte and Rondeau, 2005). Maclennan et al. (2009) argue that a positive consequence of a compensation program in southern Kenya is that it has contributed to a reduction in the number of retaliatory killings of lions. On the negative side, several studies note that compensation schemes reduce private incentives to protect livestock (Bulte and Rondeau, 2005; Zabel et al., 2011). Rollins and Briggs (1996) therefore suggest that depredation payments should be conditional on observed abatement efforts by farmers. Moreover, if compensation levels are set at a high level, compensation can have an effect similar to that of subsidies to farming and livestock, providing incentives for entry into the sector and thus increasing the number of producers operating in the area (Rollins and Briggs, 1996; Nyhus et al., 2005; Bulte and Rondeau, 2005). This can increase wildlife damages, if it increases the amount of prey available to predators. However, if livestock is free-ranging, an increase in livestock can lead to stronger competition between livestock and wildlife for food and space, which can reduce stocks of wildlife (Bulte and Rondeau, 2005). Also, if compensation schemes imply that labor is reallocated from defensive hunting to farming this could lead to further decreases in wildlife habitats (Rondeau and Bulte, 2007).

Rollins and Briggs (1996) note that a public compensation scheme could, at least hypothetically, function as an insurance scheme, where the government faces a problem similar to that of an insurance company. However, few studies have investigated private insurance schemes against wildlife damages. An exception is Mishra et al. (2003), that report on a program intended to provide incentives to manage snow leopard depredation, and where farmers are asked to give up a share of grazing land to ensure the availability of natural prey to the snow leopard, while receiving compensation for land abandoned. This compensation is the used to create a communal insurance fund to offset costs of livestock losses.

An alternative to compensation for killed livestock is to condition payments on the abundance of wildlife on the land owner's property, so called performance payments (Ferraro and Kiss, 2002). The only program currently in operation is that in Sweden for lynx and wolverine populations maintained on reindeer herders' land (Zabel et al., 2011). Performance payments are paid in advanced, based on the expected offspring, and is supposed to cover the expected damage cost. Such programs are proven to be more cost-effective than ordinary compensation programs, and Zabel et al. (2011) show that the relative cost of ordinary compensation and performance payment programs depends on the relationship between predator and prey. However, the policy requires identification of eligible groups as well as a mechanism for distributing the compensation among individuals in affected groups (Zabel and Engel, 2010; Zabel et al., 2011). Similarly to compensation schemes, performance payments create incentives for reduced poaching and increased entry in the affected sector, but different to compensation schemes it does not reduce incentives for preventive efforts which reduce the wildlife damage (Zabel et al., 2011). In addition, transaction costs for such ex ante compensation schemes can be lower than for ex post compensation schemes (Schwerdtner and Gruber, 2007).

4.3 Policy instruments for abatement of wildlife damage

Policy instruments are used to provide incentives for abatement of wildlife damages. Abatement methods that reduce the damage caused by wildlife can be graded according to the harm caused on the animals (Rollins and Briggs, 1996; Yoder, 2000), where hunting is the most harmful abatement method. Public policies that control abatement include regulation of allowed hunting pressure and incentives for increased hunting, as well as incentives for non-lethal abatement methods (Rollins and Briggs, 1996) such as; translocation of problem animals, scare devices, guard dogs, barriers or improved livestock husbandry (Breck and Meier, 2004). Abatement strategies for reducing damages have been examined by means of cost-benefit analysis, e.g., for supplemental feeding programs for black bears taking into account the reduced forestry damages (Ziegltrum, 2004), and cost-efficiency analysis, e.g. with regard to the trade-off between fox culling and various prevention measures for sheep farms (Moberly et al., 2004). In the latter case, it is shown that the optimal solution varies across farm type and location, implying that uniform policy can be expensive compared to a differentiated one. As observed by Rollins and Briggs (1996), voluntary subsidies to farmers for prevention of wildlife damages can give rise to spatial externalities; an increase in prevention effort by an individual farmer can imply that neighboring farmers experience an increase in depredation, and at worst there is no net gain to society.

Few have studied the role of wildlife damage policies for the economic viability of livestock sectors. An exception is Berger (2006), that evaluates the impact of predator culling subsidies 1939-1980 on sheep industry size, and shows that the impact of the subsidies was small compared to that of output prices and production costs. The author therefore argues that direct compensation is more efficient than predator control in ensuring the profitability of the sheep industry (Berger, 2006).

In addition to the economic incentive schemes provided by governments, there are substantial voluntary payments to individuals or groups for supplying wildlife or wildlife habitats (Ferraro and Simpson, 2002). Such schemes operate both within countries, and across country borders. Much of these payments are channeled through eco-friendly “products”, such as eco-tourism, but Ferraro and Simpson (2002) argue that paying directly for ecosystem habitat protection would be more cost-effective. For example, direct subsidies to rain forest preservation would be a more efficient way to protect rain forest habitats compared to subsidies to investments or production in the bee-keeping sector, even though bees contribute to habitat quality.

5. Summary and conclusions

The aim of this paper was to provide an overview of studies calculating benefits and costs of wildlife, and of studies on policy design for efficient wildlife management. To this end, we reviewed 36 studies on benefit and 37 on cost estimates, which were found in scientific journals and in the ‘grey’ literature which includes reports from authorities and consultancy firm. They were identified by defining key words ‘wildlife’, ‘benefits’, and ‘costs’ at different search engines such as Web of Science, Scopus, and Google scholars. It was found that the literature on the calculation of both benefits and costs can be dated back to the early 1970s, and that the results differ among studies in both categories.

With respect to benefit estimates, recreational value from hunting was the most commonly estimated benefit type, and the results varied between13 and 545 USD/hunting day, and between 18 and 225 USD/hunting trip as measured in 2013 prices. The literature applied two main methods, stated and revealed preference methods, in equal proportions on the estimation of hunting values. There was a tendency to move from contingent valuation to choice experiment over time as the most used stated preference method. The favorite animals were deer, moose, and elk, which accounted for 65% of all studies. A majority of the studies, 70%, were applied to US, and the rest to Europe and Australia. We did not find any study on hunting values in developing countries. This is in contrast to studies estimating regional economic effects of wildlife, where 4 out of 9 studies were applied to countries in Africa. These effects were estimated as expenditures on equipment, housing and other for the hunters and in 2 studies, applied on wildlife in the US, dispersal effects in the rest of the economies. However, Hoves (2002) showed that only direct expenditures can correspond to 0.5% of GDP in some European countries

The 37 studies estimating costs of wildlife were focused on cost of livestock predation, crop destruction, traffic collision, and transmission of diseases to animals and humans. The major part of the literature, 20 studies, estimated costs of carnivores' predation on livestock followed by another part of the literature, 11 studies, estimating costs of crop losses. For both these items, costs were calculated by means of data on market value of livestock/crop or compensation payments for losses. Costs of crop losses were also estimated by questionnaires to farmers on their perceived losses. For comparable estimates, the results range between 18 and 6620 USD/household/year for livestock losses and between 0-2630/household/year for crop losses. These estimates can correspond to approximately 40% reduction in profits for individual farmers in wildlife rich regions in the US. Deer and wild boar are the main agents of crop losses, and a mix of different carnivores for livestock predation. It is interesting to note the relatively large number of studies in developing countries, 11 out of 31, compared with studies on wildlife benefits. One explanation is the costs of livestock predation born by neighboring farmers by the establishment of protected areas in several developing countries.

Most of the reviewed studies calculated either one type of benefit or cost usually for one animal. As shown in current paper, the benefit and costs are expressed in different ways and are difficult to compare and, hence, make conclusions on whether wildlife benefits exceed costs. In order to make such conclusions both costs and benefits may need to be related to wildlife population levels, which is a challenging task. Skonhoft (2006) carried out such analysis for wolf in Scandinavia and showed that the benefit/cost ratio is heavily dependent on the predation on moose which create costs from traffic collisions. However, other costs for the farmers in addition to the loss of livestock/crop and dispersal effects in the economy are generally not included, which would raise the cost. On the other hand, benefits associated with wildlife contribution to biodiversity and associated effects on provision of ecosystem services have not been considered. Thus, a full-fledged cost benefit analysis of single or several animals remains to be carried out.

The literature on policies for wildlife management identified three potential mechanisms for solving conflicts; distribution of property rights, protection by law of endangered wildlife species, and compensation payments. A common agreement is that the first two mechanisms will not function in isolation since this will give incentives to those who experience costs of wildlife to decrease the population by illegal hunting and killing. Compensation payments, which can be raised by taxes of hunting fees, are then needed. These payments can be based on stated losses, market values of livestock/crop, or performance of wildlife. Compensation payments based on stated damage cost might be too high since there is an incentive to overstate the actual costs of the loss. An alternative is then to base the payment on the market value of the livestock/crop loss. This will not account for other costs affecting the farmers, such as preventive measures. Common to both these payments is that they create disincentives to invest in preventive measures. Performance payments are then preferred, but are difficult to implement and enforce.

We found that a considerable number of the papers on policy design analyzed the implications of ownership rights and how those are distributed across different groups affected by wildlife, and across space. However, the efficiency of private property rights in managing wildlife is likely to be strongly dependent on the functioning of markets for the benefits and damages provided by the wildlife, and few have analyzed these markets. In that regard, there is a parallel to the relatively abundant literature about wildlife damage compensation, as there seems to be almost no analysis of the prospects for, or existing, private insurances against wildlife damages. An alternative not considered in the literature is the design of liability rules for occurrence of damage. Such a system is implemented in Germany, where the holders of hunting licenses are responsible for damages caused by wildlife for the land owner. There is also a relatively small literature on both evaluation of actual policies for abatement of wildlife damages, and analysis of optimal policies and how they should be applied given heterogeneous conditions across space and wildlife species migration.

In sum, the main conclusions from our survey are that a relatively large body of literature has developed since 1970s on the estimation of costs and benefits of wildlife, which has been applied mainly on single animals in the developed economies, in particular in the US. However studies concerning the indirect impact and cost of predation could benefit from more attention. Correspondingly, there is a lack of studies comparing costs and benefits in a wider setting including several costs and benefits items, which can be compared. Such studies would provide important basis for the design of policies, the literature of which is comprehensive and thorough but lack empirical applications on interdependent wildlife populations and their interactions with humans. Developments are also needed with respect to evaluations of the functioning of the many existing policies in order to improve and fine tune their efficiency.

Appendix: Tables A1-A7

Table A1: Direct expenditures on hunting and share of GDP (gross domestic product) in some European countries

Country	Annual direct expenditure on hunting, million USD[1]	Share of GDP in 2001, in %[2]
Austria	209	0.11
Belgium	411	0.18
Finland	302	0.24
France	3409	0.25
Great Britain	7014	0.48
Ireland	115	0.11
Portugal	261	0.22
Spain	47	0.01
Sweden	305	0.13

[1]Hofer (2002); [2] GDP in 2001 from Nationmaster (2015)

Table A2: List of TCM/hedonic methods valuing the value of hunting (in 2013 USD)

Reference	Study Period	Subject (Animal)	Method	Region	Recretional value
Wennergren et al. (1973)	-	Deer	TCM	Utah	48.55 USD per day
Martin et al. (1974)	1970	Big game (Not deer)	TCM	Arizona	109.16 USD
Brown and Plummer (1979)	1976	All big game	Hedonic	Oregon	183.16 USD
Livengood (1983)	1978-1979	White-tailed deer	Hedonic	Texas	70.73 USD for the first deer harvested
Sorg and Nelson (1986)	1982	Elk	TCM	Idaho	72.80 USD per day
Donelli and Nelson (1986)	1982	Deer	TCM	Idaho	55.63 USD per day
Young et al. (1987)	1982	Upland game (rabbit, pheasant, quail, grouse, wild turkey, dove)	TCM	Idaho	59.03 USD per day for upland game 50.62 USD per day pheasant only
Duffield (1988)	1985	Elk	TCM	Montana	123.59 USD per day
McCollum et al. (1990)	1986	Big Game	TCM	U.S/Region 6	10.66 USD
Bergstrom and Cordell (1991)	1987	Big (Small) Game Hunting	Multisite TCM	U.S	18.62 (18.48) USD
Cooper and Loomis (1991)	1987-1988	Waterfowl	TCM	San Joaquin Valley	97.38 USD per day
Offenbach and Goodwin (1994)	1986	All game	TCM	Kansas	233.15 – 256.00 USD per trip
Whitten and Bennet (2001)	2000	Duck	TCM	Australia	23.47- 37.95 USD per trip
Knoche and Lupi (2012)	2003	Deer	TCM	Michigan	89.76 USD per year (for firearms-hunter) 96.01 USD per year (for archery-hunter)
Knoche and Lupi (2013)	2008	Grouse	TCM	Michigan	255.13 USD per year

Table A3: List of CV methods (except CE) valuing the willingness-to-pay for a hunting day, in 2013 USD

Reference	Study Period	Subject (Animal)	Method	Region	Willingness to pay per hunting day
Hansen (1977)	1975	Elk	CVM	Intermountain	77.20 per day
Charbonneau and Hay (1978)	1976	All big game	CVM	U.S	175.00 per day
Cory and Martin (1985)	1981	Elk	CVM	Arizona	128.69 per trip (no daily value available)
Sorg and Nelson (1986)	1982-1983	Elk	CVM	Idaho	37.80 per day (1982) 45.14 per day (1983)
Donelli and Nelson (1986)	1982-1983	Deer	CVM	Idaho	39.73 per day
Young et al. (1987)	1982	Upland game[1]	CVM	Idaho	46.50 per day for upland game 44.86 per day pheasant only
Hay (1988)	1985	Elk	CVM	Several US states	52.75 - 93.77 per day
Loomis et al. (1988)	1986	Elk	CVM	Montana	72.31 per day
Conelly and Brown (1990)	1985	Value for Federal Land Users	CVM	Several states in US	53.33-86.08 per day
Mackenzie (1990)	1989	Deer	Conjoint analysis	Delaware	1133.49 USD statistical value per deer[2]
Gan and Luzar (1993)	1990-1991	Waterfowls	*CVM (conjoint analysis)*	Lousiana	612.12 US-$ per hunting seasons and given attributes
Morton et al. (1994)	1991	Deer, Moose	CVM	Saskatchewan, Kanada	23.93–78.34 US-$ per hunting trip for deer 40.25–183.93 US-$ per hunting trip for moose
Fried et al. (1995)	1989-1991	Elk	CVM	Eastern Oregon	84.96 per day
Boman et al. (2011)	1987, 2006	Moose	CVM	Sweden	106 US-$(1987) 91 US-$ (2006)
Boman and Mattson (2012)	1987, 2006	Moose	CVM	Sweden	+ meat value

[1]rabbit, pheasant, quail, grouse, wild turkey, and dove; [2]Not a WTP . The value of a marginal increase in the probability of bagging a dear is USD 11.3

Table A4: List of Choice Experiments dealing with hunting valuation in 2013 USD

Reference	Study Period	Subject (Animal)	Region	Findings/ Value (in 2013 US-$)
Bullock et al. (1998)	1996	Red deer (Stags)	Scottish Highlands	Hunting day value: 545,06 - High quality and number of deer is appreciated - Forest conservation and mixed terrain reduce the value of shooting licences of red deer
Horne and Petäjistö (2003)	2001	Moose	Finland	- Moose population over pareto-optimum, landowners would appreciate a reduction - Cost for landowners per animal: 383.09 USD - Average Bag-value per animal: 455.11 USD - Even with full participation on licence-fees, landowners would decrease population
Delibes-Mateos et al. (2014)	2012	Red-legged partridge	Spain	- Clear preference of shooting wild partridges instead of farm reared (20-times higher) - Higher WTP if additional small-game may be hunted - Hunters have a higher WTP if they hunt in ecologically better conditions WTP/wild partridge: 66.28$ WTP/farm-reared: 3.35$ (both with decreasing marginal value for subsequent shots)
Kerr and Abell (2014a)		Sika deer	New Zealand (Kaimanawa and Kaweka Forest Park)	- 125.29 US-$ per hunting day - High number of Sika deers improve the value to hunters
Kerr and Abell (2014b)		Himalayan Thar and Sika deer	New Zealand (Kaimanawa and Kaweka Forest Park)	-Values for Sika deer see Kerr and Abell (2014a) - WTP: 267.53 US-$ (2014-USD) for a hunting day on Himalayan Thar for a 5 day hunting trip (differing values depending on the length of the trip)

Table A5: Studies on direct and indirect regional economic effects of hunting

Study	Study period	Animal	Method	Region	Value in USD
Burger et al., (1999)	1991	Northern Bobwhite	I-O	Southern U.S	331.11 million
US Department of Interior, (2011)	1991-2011		CVM and expenditure	U.S	16.58-34.81 billion
Grado et al. (2007)	2001-2005	White-Tailed Deer	SAM	Mississippi	1.33 Billion (2001) – 0.833 Billion (2003)
Booth (2010)	2000-2009		Expenditure	7 countries in Africa	5.37 Million - 83.91 Million
Baumis et al. (2010)	2009		Expenditure	Latvia	44.79Million
Humavindu and Barnes (2003)	2000		Expenditure	Namibia	25.51 Million
Lindsey et al. (2006)	1995-2003		Expenditure	10 countries in Africa	0.8 Million - 100 Million
Jorge et al. (2013)	2010	Leopard	Expenditure	Niassa National Reserve, Mozambique	25275
CREST (2014)	2012	Black and Grizzly bear	Expenditure	Great Bear Rainforest of British Columbia	Hunting Expenditure = 1.22 Million

Table A6: Studies on costs of livestock predation

Study	Period	Animal	Region	Type of cost estimate	Result in 2013 USD
Taylor et al. (1979)		Coyotes	Southwestern Utah	Direct and indirect costs	1.1 and 3.1-4.85 million
Yom-Tom et al. (1995)	1993	Golden jackal	Golan Heights, Israel	Direct, market value	59 368
Mishra (1997)	1996	Large carnivore	Indian trans-Himalaya	Direct, market value	181/household/year
Ciucci and Boitani (1998)	1992-1995	Wolf and dog	Central Italy	Direct, compen-sation payouts	487 665/year
Butler (2000)	1993-1996	Baboons, linons, leopards	Gokwe , Zimbabwe	Direct, market value	18.38 / household/year
Chardonnet et al. (2002)	1997	Bear, wolf	Austria, France, Greece, Italy, Spain	Direct, market value	0.01 – 1.5 million
Rao et al. (2002)	1996-1997	Leopard, bear	Nanda Devi, India	Direct, market value	36 700
Madhusudan (2003)	1996-1999	Large felines	Bhadra Tiger Reserve, India	Direct, market value	75/ household /year
Ikeda (2004)	2001	Snow leopard	Kanchenjunga, Nepal	Direct, market value	69 - 515/ year/household
Asheim and Mysterud (2004)	1992-1993	Carnivore	Norway	Direct, market value and other costs	4.42-19.01 million (77% market value)
Jones (2004)	1999	Coyotes, dogs, mountain lions	U.S	Direct and indirect, I-O	21 and 16 million
Michalski et al. (2006)	2001-2004	Large felines	Brazil	Direct, market value	326-33315
Gusset et al. (2009)	2005	wild dogs	Northern Botswana	Direct, compen-sation payouts	174 000
Tamang and Bara (2008)	1993-1998	Large cats	Bardia, Nepal	Direct, market value	15800
Sommers et al. (2010)	1990-2004	Large carnivores	Wyoming	Direct, uncompen-sated loss	234 000
NASS U.S, (2010)	2009	Carnivores	U.S	Direct, market value	22 million
Abaya et al. (2011)	2005-2009	Spotted hyenas	Northern Ethiopia	Direct, market value	7 322
NASS, U.S, (2011)	2010	Carnivores	U.S	Direct, market value	104 million
Rashford (2010)		Carnivores	U.S.	Direct, market value and others	40% decrease in profits
Steele (2013)		Wolf	U.S.	Direct, market value and others	223000 and 349000-540000
Ramler et al. (2014)		Carnivores	U.S.	Direct, market value and others	890 and 6620/farm/year

Table A7: Studies on costs of wildlife damages on crops, in 2013 USD

Study	Period	Animal	Region	Type of damage	Results in 2013 USD
Brodie et al. (1979)	1963-1975	Wildlife	Oregon and Washington	Market value of coniferous plantation	0.601-4.79 billion
Vogel (1989)		Deer	Pennsylvania	Perceived loss of crops	59.53-142.21 million or 1112-2630/household/year
Heinrich and Craven (1992)	1985-1986	Canada Geese	Horicon Marsh, Wisconsin	Perceived loss of crops	2.41 million
Wywialowski (1994)	1989	Deer, coyotes	U.S	Percieved loss of crops	289-433 /household/year
Conover (1994)	1993	Deer, coyotes	U.S	Percieved loss crops	0-<1444/household/year
Garshelis et al. (1999)	1991	Black bear	U.S.	Perceived loss of crops	2038/household/year (in 1991)
Frederick (1998)	1996	Wild boar	California	Perceived loss of crops	2.3 million
Chardonnet et al. (2002)	2000	Wild Boar	France	Compensation payment for crop/forestry damage	38.4 million
Rao et al. (2002)	1996-1997	Wild boar, bear, porcupine	Nanda Devi biosphere reserve, India	Market value of crops	19 300
Madhusudan (2003)	1996-1999	Elephants	Bhadra Tiger Reserve, India	Market value of crops	144 /household /year
Hofman-Kaminska and Kowalczyk (2012)	2000-2010	European Bison	Northeastern Poland	Compensation payment for lost crop	274 000

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